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Carbon footprint of different agricultural systems in China estimated by different evaluation metrics

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ABSTRACT

Increase of concentrations of greenhouse gases (GHGs) in atmosphere have garnered lots of attention due to the associated risk of global warming. Carbon footprint (CF) is a measure that is extensively used to assess the contribution of an individual, event, organization, or product to global climate change. In the current study, the CF of five agricultural systems with different crop systems in different regions in China was evaluated in terms of CO_2 eq per hectare, per kg crop yield, and per unit economic output, association with soil GHGs, agrochemical production and soil organic carbon (SOC) change. The results showed that direct nitrous oxide accounted for 61.9–97.8% of total soil GHG emissions in upland fields (agroforestry system, summer maize-winter wheat rotation tillage system, maize-soybean intercropping system, and continuous maize tillage system), while soil methane accounted for 89.9–95.1% of total soil GHG emissions in the paddy field (double rice-winter crop rotation system). GHGs produced from agricultural inputs varied from 2698.7 to 9781.5 kg CO_2 eq ha^{-1} yr^{-1} among different agricultural systems. Nitrogen fertilizer was the dominant source of GHGs, which accounted for 47.4–92.4% of the total emission. Annual SOC increase ranged from −639.9–13823.9 kg CO_2 eq ha^{-1} yr^{-1} in the five agricultural systems. Increases of SOC in the treatment of "reduced tillage with straw return" offset all of the total GHG emissions while "conventional tillage without straw return" decreased SOC storage in the continuous maize tillage system. An increase of SOC in crop systems of the other four agricultural systems could offset 23.2–66.7% of the total GHG emissions. Carbon emissions per hectare in the double rice-winter crop rotation system (7339.93–22994.17 kg CO_2 eq ha^{-1} yr^{-1}) were higher than the other four agricultural systems (−744.96–6523.12 kg CO_2 eq ha^{-1} yr^{-1}). In terms of crop yield, CFs of the summer maize-winter wheat rotation tillage system (0.28–0.34 kg CO_2 eq kg^{-1} yr^{-1}) were lower than those from the other four agricultural systems (0.39–2.83 kg CO_2 eq kg^{-1} yr^{-1}), except for the treatment with reduced tillage and residue return in the continuous maize tillage system. Similarly, CFs, as quantified via per unit economic output of the agroforestry system (0.12–0.19 kg CO_2 eq $¥^{-1}$ yr^{-1}), were lower than those in other the four systems (0.26–0.89 kg CO_2 eq $¥^{-1}$ yr^{-1}).Different evaluation methodology greatly affected the CF of different crop systems and mostly likely contributed to the contrary evaluation results, especially in the complex system of various crop systems (agroforestry and double rice-winter crop rotation system) in current study. Evaluation results of the CF of an agricultural system depends highly on the methodology used and may suggest different management decisions for lowering CF to promote the development of cleaner production.

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1. Introduction

Increasing atmospheric concentrations of GHGs are of great concern owing to the related risk of global warming. Agricultural ecosystems are a major source of GHGs (mainly CH_4:methane; CO_2:carbon dioxide and N_2O: nitrous oxide), which reaches a net emission of 6.1×10^9 t CO_2 eq. yr^{-1}(approximately 12% of global emissions) (IPCC, 2007; Linquist et al., 2012). Nevertheless, there are opportunities to reduce GHG emissions by improving agricultural management because the agricultural ecosystem is the one most affected by human activity among all of the ecosystems (Gan et al., 2012; She et al., 2017). Agricultural ecosystems contribute a considerable share of GHG fluxes and concurrently act as a CO_2 sink (Al-Mansour and Jejcic, 2016). According to the results reported by IPCC (2014), the global economic mitigation potentials for agriculture in 2030 are estimated to reach up to $0.5–10.6 \times 10^9$ t CO_2 eq yr^{-1}.

To measure the GHG emissions intensities related to human activity in different scenarios, the term of carbon footprint (CF) could be used as an indicator (Gan et al., 2011a). For instance, Rashid et al. (2018) reported the CF of living style standards of urbanized areas of Rawalpindi was significantly higher than national standard values in Pakistan. They encouraged adopting less energy-wasting habits and using more green resources to decrease stress on the ecosystem in order to reduce GHG emissions. Vargas-Ramirez et al. (2017) reported that CF of conventional fermentable sugars of corn grain and sugar beets would be decreased by 52% and 29% by improving storage technics, respectively. Gan et al. (2011b) found that CF of durum grown in a pulse-pulse-durum system was 34% lower than that in cereal-cereal-durum systems. The CF of durum could be lowered by diversifying cropping system.

China is a large agricultural country. It produced grain for 22% of the world's population with 7% of the total global arable land (Wang, 2016). For decades, various intensive agricultural practices with high agricultural inputs have been developed to maintain crop yield in China. Consequently, GHGs emitted from agriculture reached 17% of the total emissions. Nevertheless, low carbon techniques have been encouraged in recent year (Dong et al., 2008; She et al., 2017). To take advantage of these opportunities, the assessments of CF of different agricultural ecosystems in China are of vital importance. Currently, different functional units could be used simultaneously when assessing the CF of an agricultural system. For example, Knudsen et al. (2014) determined the CF at the farm gate from the full crop rotations with two functional units (per hectare and per kg total yield) in different rotation systems. Similarly, these two functional units were used in other studies (Rakotovao et al., 2017; Xu and Lan, 2017). Yang et al. (2014) estimated the CF in different rotation systems with three functional units (per unit area, per kg total biomass, and per unit economic output). The difference in CF of different single crops (e.g., rice, maize, wheat, and potato) from different regions in China had been evaluated with the functional units of per hectare and per kg total yield (Xu and Lan, 2017; She et al., 2017). However, studies with multiple analyses of CF in different agricultural systems with various crop systems within different regions in China are rarely reported. Furthermore, the multiple analyses of CF in different agricultural systems may produce contradictory results (Knudsen et al., 2014) and might suggest different strategies for reducing CF. This raises the question of which evaluation metric is most suitable for estimating the CF among different agricultural systems and producing reliable results. Therefore, the objective of the current study is to determine and compare the annual CF of different agricultural systems with various crop systems in five different regions of China in terms of per hectare, total yield of crops, and economic output. The analysis will explore differences between various methods for evaluating the CF of different agricultural systems with various crop systems in China, and the results will provide a basis for making policies to promote the development of low carbon agriculture in China.

2. Sites, materials and methods

2.1. Experimental site

Normally, China is divided into six agricultural regions: northeast, North China Plain, northwest, middle-lower reaches of Yangtze River, southwest, and South China (She et al., 2017). In the current study, four agricultural regions were included: the northeast, North China Plain, middle-lower reaches of Yangtze River, and South China. Maize, maize and wheat, rice, and rubber are the dominant crops in these regions, correspondingly. Five different agricultural systems that include these dominant crops were used to estimate CF, i.e., rubber-crop agroforestry system, double rice-winter crops one-year rotation system, a summer maize-winter wheat one-year rotation tillage system, a maize-soybean intercropping system, and a continuous maize tillage system. Detailed information about the study sites for the five agricultural systems is shown in Table 1.

Table 1
Study site information of different agricultural systems.

Agricultural system	Location (city, province)	Latitude and longitude	Basal soil property	Climate	Mean annual temperature and precipitation
Trial 1: Agroforestry system	Danzhou, Hainan	19.55°N, 109.48°E	Texture: silty clay loam; SOC: 7.1 g/kg; AP: 31.5 m g/kg; AK: 56.4 mg/kg; pH: 4.5	Tropical monsoon	23.4 °C; 1826 mm
Trial 2: Double rice-winter crop rotation system	Changsha, Hunan	28.14°N; 116.57°E	Texture: clay loam; SOC: 19.7 g/kg; AP: 84.0 mg/kg; AK: 87.0 mg/kg; pH: 5.4	Subtropical tropical monsoon humid	17.1 °C; 1500 mm
Trial 3: Maize-wheat rotation tillage system	Yucheng, Shandong	36.84°N; 113.20°E	Texture: silt loam; SOC: 7.5 g/kg; AP: 21.2 mg/kg; AK: 113.7 mg/kg; pH: 8.6	Temperate semiarid	13.4 °C; 567 mm
Trial 4: Maize-soybean intercropping system	Beijing	39.95°N, 116.30°E	Texture: sandy loam; SOC: 2.9 g/kg; AP: 12.3 mg/kg; AK: 45.7 mg/kg; pH: 8.1	subhumid temperate monsoon	11.0 °C; 600 mm
Trial 5: Continuous maize tillage system	Gongzhuling, Jilin	43.75°N, 125.01°E	Texture: clay loam; SOC: 13.2 g/kg; AP: 30.7 mg/kg; AK: 115.2 mg/kg; pH: 6.5	Temperate monsoon	5.6 °C; 562 mm

SOC: soil organic carbon; AP: available phosphorous; AK: available potassium.

2.2. Experimental design

2.2.1. Trial 1: agroforestry system

The experimental site of trial 1 is located in an experimental farm belonging to the Chinese Academy of Tropical Agricultural Sciences in the city of Danzhou, which is the largest rubber production area in Hainan Province, China. This experiment consisted of four treatments: conventional rubber plantation of single row system (RPSR), rubber plantation of double row system (RPDR), rubber plantation of double row system (RPDR), RPDR intercropping with yam bean and peanut (RPDR-Y and RPDR-P), respectively. The rubber tree (*Hevea brasiliensis*) is a rubber-producing arbor. Trees in RPSR was planted at the recommended density (7 m × 3 m, 480 plants ha^{-1}), while RPDR which was adjacent to RPSR, was planted at an inter-row spacing of 20 m, intra-row spacing of 4 m, and a tree spacing of 2 m (420 trees ha^{-1}) in 2002. The harvesting of latex from the rubber trees commenced on August in 2010, the time at which the tree reached maturity. The total rubber planting area was 6 ha (RPDR: 4 ha; RPSR: 2 ha). Latex was tapped once every three days from April to December. After harvesting, latex was transported to a factory by truck. In mid-February or late February, the leaves of the rubber tree fall in short term, and then buds appear soon after in early March when air temperature increases rapidly in this area. The leaves that had completely fallen off the tree from the two rubber plantations (21 m^2 for RPSR and 24 m^2 for PRDR with three replications, respectively) were collected to determine the return of biomass from the rubber trees in late February in 2016 and 2017.

At maturity, the RPSR form a continuous heavy shade, hindering the growth of lower strata crops, while RPDR is able to avoid heavy shade, thereby making intercropping possible. The annual legume crops, yam bean (*Pachyrhizus erosus*), and peanut (*Arachis hypogaea Linn.*), were planted in the central 12 m section of the 20-m wide inter-row of RPDR with three replications (50% of land was used for intercropping with annual crops). The area of the intercropped plot was 16 m × 12 m. The yam bean and peanut crops were sown on April 1 and March 31, respectively. For years 2014 and 2015, yam bean was harvested on August 15 and August 31, respectively, whereas peanut was harvested on July 2 and July 7, respectively. The plant residue of the harvested yam bean and peanut was returned to field after harvest.

2.2.2. Trial 2: double rice-winter crop rotation system

The experimental site of trial 2 is located at the experimental station of the Institute of Soil and Fertilizer Research, Hunan Academy of Agricultural Sciences, in the city of Changsha, Hunan province, China. This experiment included four one-year cropping systems: typical double rice (traditional early and late rice) with winter fallow (DR), double rice-rape (DR-Ra), double rice-potato (DR-P) and double rice-Chinese milk vetch (DR-C). Normally, early rice and late rice were planted in mid-April and early July, respectively. Winter crops were planted in October after the harvest of the late rice. The area of each plot was 32 m^2. The crop residue was returned to the field after harvesting. All of the biomass from the Chinese milk vetch was returned to the field as green manure before the seeding of early rice. Details of the experimental design were reported in previous studies (Tang et al.,2010, 2011; Huang et al., 2011a).

2.2.3. Trial 3: maize-wheat rotation tillage system

The experiment of trial 3 was conducted at the Yu Cheng Comprehensive Experimental Station of the Chinese Academy of Sciences in Yucheng in Shandong Province in the North China Plain. The typical cropping system in current area is summer maize-winter wheat rotation. In each crop rotation year, summer maize was sown in June and harvested in October, whereas winter wheat was sown in October and harvested in June next year. This experiment included two treatments of conventional tillage (CT) and no-till (NT). Plots of CT treatment were ploughed to a depth of 30–35 cm approximately a week before sowing, while those in NT treatment were not disturbed. The area of each plot was 300 m^2. In CT and NT, the residue and inorganic fertilizers were spread onto the soil surface. Details of the experimental design are the same as those previously reported by Zhang et al. (2018).

2.2.4. Trial 4: maize-soybean intercropping system

The field site for trial 4 was located at the Shangzhuang Experimental Station of China Agricultural University in Beijing, China. This experiment included two treatments of monoculture maize (M) and maize-soybean intercropping (MS). Maize was planted at an 80-cm inter-row spacing and an 40-cm intra-row spacing in the two treatments. In MS, soybean was sown within inter-row spacing (two rows of maize and one row of soybean). Maize and soybean were manually sown in mid-June and harvested mid-September in 2010 and 2011. The area of each plot was 24 m^2. No pesticides or herbicides were used in the field. A no-tillage treatment was used in current study. Most residues were manually removed after harvest. Detailed experimental design was reported in previous study (Huang et al., 2014).

2.2.5. Trial 5: continuous maize tillage system

The experiment for trial 5 was conducted at the Gongzhuling Conservation Tillage Base of the Jilin Academy of Agricultural Sciences in the city of Gongzhuling in Jilin Province, China. The typical cropping system in this area is continuous maize. This experiment included two treatments of “conventional tillage without straw return” (CTWS) and “reduced tillage with straw return” (RTS). Maize was planted in April or May and harvested in October before the arrival of the winter coldness. In CTWS, field plots (plot size: 110 m^2) were ploughed before seeding. Maize was mechanically planted at a row spacing of 65 cm. After harvesting, most residues were mechanically removed, except for the stubble (15 cm height). In RTS, maize was planted at an 90-cm inter-row spacing and an 40-cm intra-row spacing. Base fertilizers were applied during seeding. A top dressing of fertilizer was applied to the maize when operating deep scarification in CTWS and in the inter-rows of RTS. In the following year, maize was mechanically sown in the previous year’s inter-row spacing, and the previous year’s intra-row spacing became the current year’s inter-row spacing in RTS. Details of the experimental design are the same as those previously reported by Huang et al. (2011b).

2.3. Data collection

2.3.1. Output and input

Crop yield, economic output, amount of residue return, and agricultural inputs, including fertilizer, pesticide and herbicide, electricity for irrigation, diesel for seeding, tillage, harvest, and transport of product, are shown in Table 2. Detailed information is described below.

In trial 1, annual rubber yield was collected from the two rubber plantations. Average annual rubber yield was calculated from 2011 to 2016. The fresh commercial tubers of the yam bean and peanut in three intercropped plots along with aboveground biomass were harvested and weighted in 2014 and 2015, respectively. Samples of aboveground biomass of the two crops were dried at 85 °C to determine the amount of residue return. Peanut samples were sun-dried to determine the yield of peanut without shell. The average yield and amount of residue return of the two crops are shown based on the average value of the two years’ date. The type and

Table 2
Crop yield, economic output, amount of residue return, and agricultural inputs of different crops in the five agricultural systems.

Agricultural system	Treatment	Crop	Economic output	Residue return	Yield	N	P_2O_5	K_2O	Diesel	Pesticide	Herbicide	Electricity
			(¥ ha^{-1})	(kg ha^{-1})								(kwh ha^{-1})
Trial 1: agroforestry system	RPSR	Rubber	25596.3	3251.2	1505.8	201.6	124.8	105.6	86.6	–	0.9	–
	RPDR	Rubber	21305.3	4118.0	1253.5	176.4	109.2	92.4	86.6	–	0.7	–
	RPDR-Y	Rubber	21305.3	4118.0	1253.5	176.4	109.2	92.4	86.6	–	0.7	–
		Yam bean	21027.4	410.0	7009.1	125.0	50.0	45.0	2.5	–	0.8	–
	RPDR-P	Rubber	21305.3	4118.0	1253.5	176.4	109.2	92.4	86.6	–	0.7	–
		Peanut	2632.1	1596.4	628.0	90.0	50.0	45.0	2.5	–	0.8	–
Trial 2: double rice-winter crop rotation system	DR	Rice	26344.7	19500.0	13919.3	513.0	90.0	225.0	175.0	19.1	3.0	1409.9
	DR-Ra	Rice	27792.6	19500.0	14684.3	513.0	90.0	225.0	187.5	19.9	3.0	1409.9
		Rape	11490.5	7500.0	2249.9	185.4	43.2	–	–	–	0.8	–
	DR-P	Rice	28161.5	19500.0	14879.2	513.0	90.0	225.0	187.5	29.5	2.3	1409.9
		Potato	26401.3	6000.0	22498.9	153.0	30.0	36.0			2.3	–
	DR-C	Rice	26827.3	19500.0	17429.1	513.0	90.0	225.0	168.7	13.2	3.0	1409.9
		Chinese milk vetch	–	22500.0	22500.0	75.0	–	–	–	–	0.8	–
Trial 3: maize-wheat rotation system	CT	Maize	6561.8	6000.0	8266.7	116.0	178.0	122.0	64.7	7.3	–	–
		Wheat	9459.4	4000.0	5833.3	116.0	178.0	122.0	67.3	6.8	–	867.4
	NT	Maize	6379.7	6000.0	8216.7	116.0	178.0	122.0	29.5	7.3	–	–
		Wheat	6094.6	4000.0	3758.3	116.0	178.0	122.0	29.5	6.8	–	867.4
Trial 4: maize-soybean intercropping system	M	Maize	3811.9	1763.9	4815.9	350.0	75.0	120.0	–	–	–	67.5
	MS	Maize	4315.2	1996.8	5451.8	233.3	50.0	80.0	–	–	–	67.5
		Soybean	2103.8	18.4	553.6	116.7	25.0	40.0	–	–	–	–
Trial 5: continuous maize system	CTWS	Maize	8763.8	529.8	11072.0	3448.5	270.3	10.6	87.2	–	2.5	–
	RTS	Maize	9888.8	11956.1	12493.3	3789.2	232.2	46.4	77.0	–	3.0	–

RPSR: rubber plantation of single row system; RPDR: rubber plantation of double row system; RPDR-Y and RPDR-P: RPDR intercropping with yam bean and peanut, respectively; DR: double rice-fallow; DR-Ra, DR-P, and DR-C: double rice-rape, double rice-potato, and double rice-Chinese milk vetch rotation, respectively; CT: conventional tillage; NT: no-till; M: monoculture maize; MS: maize-soybean intercropping; CTWS: conventional tillage without straw return; RTS: reduced tillage with straw return.

amount of agricultural inputs were recorded. In trials 2 and trial 5, data from the long-term experiments was provided by cooperate research teams of the Soil and Fertilizer Institute, Hunan Academy of Agricultural Sciences, and the Institute of Agricultural Resources and Environment, Jilin Academy of Agricultural Sciences, respectively. In trials 2 and 5, the output and input data were calculated based on the average value measured from 2005 to 2007 (Li et al., 2011a) and from 2001 to 2008 (Huang et al., 2011c), respectively. In trial 3, available information (from 2004 to 2009) was adapted from the literature reported by Zhang et al. (2018). Amount of diesel consumption, electricity, and pesticide usage were obtained from the same crop system in this region (Han et al., 2010). In trial 4, the output and input dates were calculated based on the average value measured in 2010 and 2011. Crop economic output was calculated by its yield multiplied by its annual price for each trial.

2.3.2. SOC

In current study, change in SOC was determined by variation in SOC storage (dSOC) or estimated by residue return. Due to a lack of field-observed data, for trial 1, the annual dSOC of different treatments was estimated to be 10% of added carbon to the upland soil (Toro-Mujica et al., 2017). In trial 2, the dSOC was estimated to be 22.44% of the added carbon to the soil in the paddy field in this region (Lu et al., 2009). This parameter is similar to other study (21.3%; Petersen et al., 2013). It is assumed that biomass contains 40% carbon (Dubey and Lal, 2009). In trials 3, 4, and 5, SOC storage in the topsoil in 2004 and 2009, 2008 and 2011, 2001 and 2008, respectively, were determined for the calculation of dSOC. The dSOC was assessed from the value of SOC storage minus the previous value. Positive dSOC indicates to a net SOC fixation. The annualized dSOC of each trial was calculated using the dSOC divided by 6, 4, and 8, respectively.

2.3.3. Soil CH_4, direct and indirect N_2O

Soil CH_4 emissions were measured only in trial 2 because soil CH_4 emissions are negligible and contribute little in the upland areas (Ceschia et al., 2010). Soil direct N_2O emissions were determined from field experiments in trials 2, 3, and 4 by using a closed static chamber method (Tang et al., 2010, 2011; Huang et al., 2014; Zhang et al., 2018). In trials 1 and 5, it was estimated by using a modified coefficient of 1.05% multiplying the total inorganic nitrogen fertilizer input based on IPCC method (Zhang et al., 2010). Annual average field-measured soil CH_4 and direct N_2O were calculated by the data observed from 2008 to 2010, 2004–2007, and 2010–2011 in 2, 3 and 4, respectively (Huang et al., 2014; Tang et al., 2010, 2011; Zhang et al., 2018).

Soil indirect N_2O emission was calculated by using a method developed by the IPCC (2006). According to this method, the N_2O emission factor of nitrate leaching NH_3–N and NOx-N from inorganic N input were 2.25,0.1 and 0.1 g N kg^{-1} N, respcectively. In trial 2, indirect N_2O emission for nitrate leaching was not included because nitrate leach was negligible in paddy soil in this region (Tian et al., 2010).

2.4. Calculation of CF

Normally, the calculation of CF does not comprise emission from variation of SOC storage in agricultural ecosystem (BSI and Carbon Trust, 2011). Nevertheless, carbon sequestration in soil is thought to have a great potential to mitigate agricultural GHG emissions (Petersen et al., 2013). In many studies, the variation of SOC storage was taken into account when evaluating CF in agricultural systems (Ramachandra and Shwetmala, 2012; Shrestha et al., 2014; Knudsen et al., 2014; Mogensen et al., 2014; Batalla et al., 2015; de Figueiredo et al., 2017; Rakotovao et al., 2017; Toro-Mujica et al., 2017). Thus, the calculations of CF including soil carbon

Table 3
Coefficients of the GHG costs from agricultural inputs (Liang, 2009).

Item	N	P_2O_5	K_2O	Diesel	Pesticides	Herbicides	Electricity
	kg CO_2 kg^{-1}						kg CO_2 kwh^{-1}
Coefficient	10.18	1.50	0.98	3.93	18.00	18.00	0.95

change, soil GHGs, and agricultural inputs were based on the entire system and are expressed annually in the current study. In many studies, multiple functional units were used when assessing the CF of an agricultural system (Knudsen et al., 2014; Yang et al., 2014; Rakotovao et al., 2017; Xu and Lan, 2017). In the current study, three functional units were used. Specifically, the CF of different agricultural systems was estimated in terms of CO_2 eq per hectare (*CFh*; kg CO_2 eq ha^{-1} yr^{-1}), per kg crop yield (*CFy*; kg CO_2 eq kg^{-1} yr^{-1}), and per unit economic output (*CFe*; kg CO_2 eq ¥$^{-1}$ yr^{-1}). The calculation formulas are expressed as:

$$CFh = GWP_{Soil\ GHGs} + GWP_{inputs} - GWP_{dSOC} \quad (1)$$

$$CFy = \frac{GWP_{Soil\ GHGs} + GWP_{inputs} - GWP_{dSOC}}{TY} \quad (2)$$

$$CFe = \frac{GWP_{Soil\ GHGs} + GWP_{inputs} - GWP_{dSOC}}{TE} \quad (3)$$

where *CFh* is the CF of a treatment per hectare, *CFy* represents the CF of a treatment per kilogram of crop yield, and *CFe* represents the CF of a treatment per kilogram of product. $GWP_{Soil\ GHGs}$ represents the total global warming potential (GWP) from soil direct N_2O–CO_2 eq, CH_4–CO_2 eq and indirect N_2O–CO_2 eq. The CO_2 eq of N_2O and CH_4 are 298 and 25, respectively (IPCC, 2006). GWP_{inputs} includes CO_2 eq emission from agricultural inputs. Table 3 lists the coefficients of the GHG costs of different categories of agricultural inputs (Liang, 2009). GWP_{dSOC} represents the GWP of annual variation in SOC storage. The mass conversion factor is 44/12. *TY* is the total yield of harvest agricultural product. *TE* is the total economic output of crops. Negative CF indicates a net GHG fixation.

2.5. Data analysis

Descriptive statistics were conducted for the current study. The data analysis was performed with the softwares (Microsoft Word 2010 and Microsoft Excel 2010). The liner relationship between soil organic carbon sequestration rate and residue return was determined using SPSS 17.0 (SPSS, Inc., Chicago, IL, USA).

3. Results

3.1. Residue return and soil organic carbon sequestration rate (SOCSR)

The annual residue return and SOCSR of the different systems is shown in Fig. 1. Result showed that residue return and SOCSR varied from 529.8 to 42000.0 kg ha^{-1} yr^{-1} and -639.9–13823.9 kg CO_2 eq ha^{-1} yr^{-1}, respectively, among different agricultural systems. SOCSR is highly correlated to the amount of residue return ($R^2 = 0.9154$, $n = 14$, $P < 0.001$; data not shown). Crop systems in trial 2 and RTS in trial 5 had higher amounts of residue return, resulting in higher carbon sequestration potentials. The highest and lowest SOCSR were recorded in DR-C in trial 2 and CTWS in trial 5, respectively. All of the treatments had a positive SOCSR except the CTWS in trial 5.

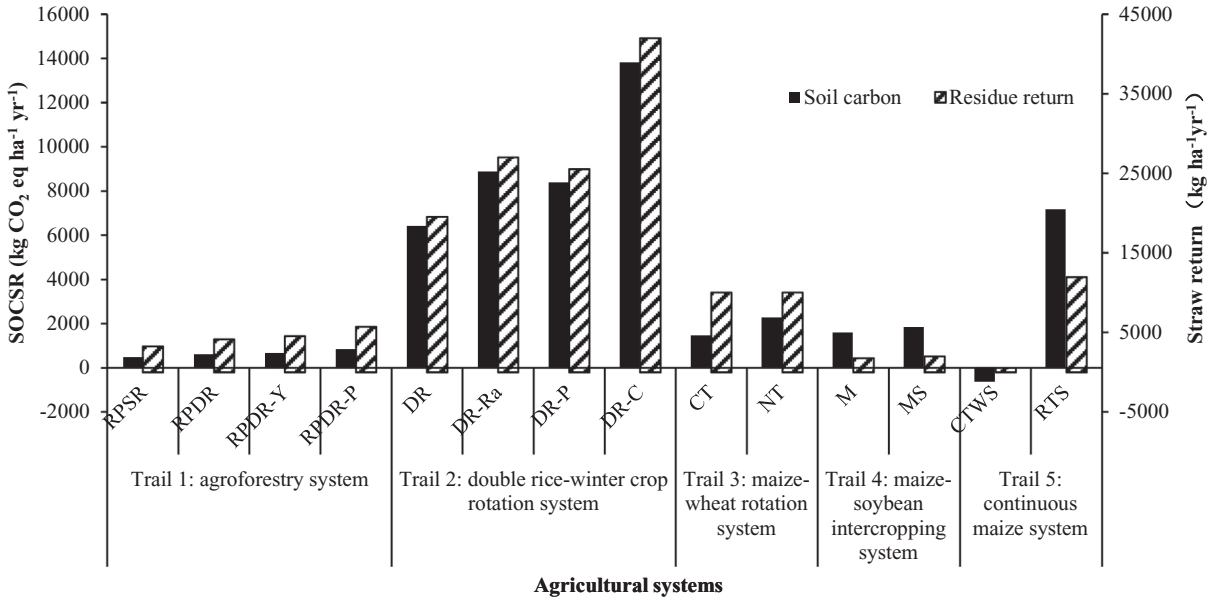

Fig. 1. Annual soil carbon sequestration rates in different agricultural systems in China. Positive values correspond to a net CO_2 fixation. RPSR: rubber plantation of a single row system; RPDR: rubber plantation of a double row system; RPDR-Y and RPDR-P: RPDR intercropping with yam bean and peanut, respectively; DR: double rice-fallow; DR-Ra, DR-P, and DR-C: double rice-rape, double rice-potato and double rice-Chinese milk vetch rotation, respectively; CT: conventional tillage; NT: no-till; M: monoculture maize; MS: maize-soybean intercropping; CTWS: conventional tillage without straw return; RTS: reduced tillage with straw return.

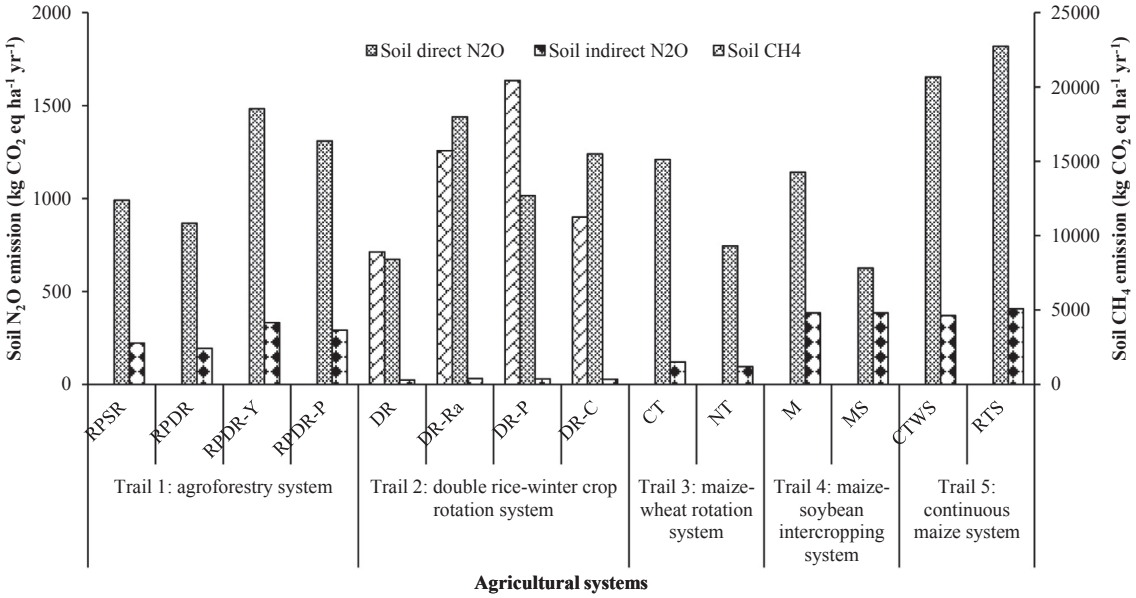

Fig. 2. Soil GHG emissions in different agricultural systems in China. RPSR: rubber plantation of a single row system; RPDR: rubber plantation of a double row system; RPDR-Y and RPDR-P: RPDR intercropping with yam bean and peanut, respectively; DR: double rice-fallow; DR-Ra, DR-P and DR-C: double rice-rape, double rice-potato and double rice-Chinese milk vetch rotation, respectively; CT: conventional tillage; NT: no-till; M: monoculture maize; MS: maize-soybean intercropping; CTWS: conventional tillage without straw return; RTS: reduced tillage with straw return.

3.2. Soil GHG emissions

Soil GHG emissions, including soil direct N_2O, soil indirect N_2O, and soil CH_4, are presented in Fig. 2. Soil direct N_2O and indirect N_2O ranged from 625.0 to 1817.9 and 24.0–406.9 kg CO_2 eq ha^{-1} yr^{-1}, respectively. In upland fields, direct N_2O accounted for 61.9–97.8% of the total soil emissions. In the paddy field, CH_4 accounted for 89.9–95.1% of total soil GHG emissions. The presence of winter crop increased the GHG emissions. In trial 2, 68.4–77.4% of CH_4 was produced in the growing season of late rice, while 40.9–58.7% and 31.7–38.9% of N_2O in growing season of winter crop/fallow and late rice, respectively were produced (data not shown).

3.3. Agricultural inputs

Fig. 3 showed that total GHG emissions from various agrochemical productions varied from 2698.7 to 9781.5 kg CO_2

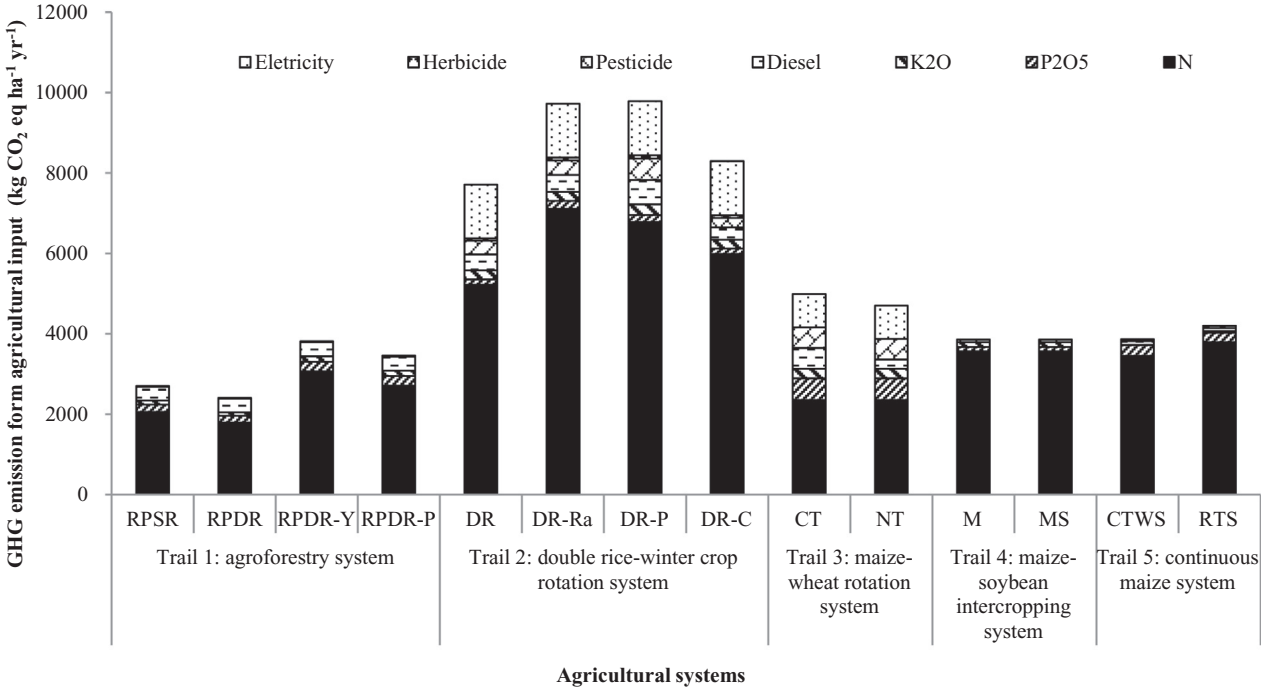

Fig. 3. GHG emissions from different agricultural inputs in various agricultural systems in China. RPSR: rubber plantation of a single row system; RPDR: rubber plantation of a double row system; RPDR-Y and RPDR-P: RPDR intercropping with yam bean and peanut, respectively; DR: double rice-fallow; DR-Ra, DR-P, and DR-C: double rice-rape, double rice-potato and double rice-Chinese milk vetch rotation, respectively; CT: conventional tillage; NT: no-till; M: monoculture maize; MS: maize-soybean intercropping; CTWS: conventional tillage without straw return; RTS: reduced tillage with straw return.

Table 4
The CF of different agricultural systems in China.

Agricultural system	Treatment	CFh (kg CO_2 eq ha^{-1} yr^{-1})	CFy (kg CO_2 eq kg^{-1} yr^{-1})	CFe (kg CO_2 eq ¥$^{-1}$ yr^{-1})
Trial 1: agroforestry system	RPSR	3307.84	2.20	0.13
	RPDR	2988.52	2.38	0.14
	RPDR-Y	5155.53	0.62	0.12
	RPDR-P	4430.77	2.83	0.19
Trial 2: double rice-winter crop rotation system	DR	11204.75	0.80	0.43
	DR-Ra	18328.53	1.08	0.47
	DR-P	22994.17	0.62	0.42
	DR-C	7339.93	0.42	0.27
Trial 3: maize-wheat rotation system	CT	4850.48	0.34	0.30
	NT	3267.23	0.28	0.26
Trial 4: maize-soybean intercropping system	M	3410.00	0.71	0.89
	MS	2638.00	0.44	0.41
Trial 5: continuous maize system	CTWS	6523.12	0.59	0.74
	RTS	−744.96	−0.06	−0.08

CFh, CFy, and CFe: carbon footprint of a treatment per hectare, per kilogram of yield and per kilogram of economic output, respectively. Negative CF indicates to a net GHG fixation. RPSR: rubber plantation of a single row system; RPDR: rubber plantation of a double row system; RPDR-Y and RPDR-P: RPDR intercropping with yam bean and peanut, respectively; DR: double rice-fallow; DR-Ra, DR-P, and DR-C: double rice-rape, double rice-potato, and double rice-Chinese milk vetch rotation, respectively; CT: conventional tillage; NT: no-till; M: monoculture maize; MS: maize-soybean intercropping; CTWS: conventional tillage without residue return; RTS: reduced tillage with residue return.

eq ha^{-1} yr^{-1} among different agricultural systems. The highest and lowest emissions were observed from DR-P and RPDR in trial 2 and trial 1, respectively. In all of the categories of input, N fertilizer was the dominant source of GHGs, which accounted for 47.4–92.4% of total emissions, while the others ranged from 0.3 to 17.4%.

3.4. CF of different agricultural systems

Table 4 presents the CFh, CFy, and CFe values for different treatments in different agricultural systems. In trial 1, RPDR had the lowest CFh indicating emission of the least GHGs of 2988.52 kg CO_2 eq ha^{-1} yr^{-1}, followed by RPSR (3307.84 kg CO_2 eq ha^{-1} yr^{-1}). The highest CFh was recorded in RPDR-Y (5155.53 kg CO_2 eq ha^{-1} yr^{-1}). RPSR also had the lowest annual CFe of the four treatments (0.13 kg CO_2 eq ¥$^{-1}$ yr^{-1}), followed by RPDR (0.14 kg CO_2 eq ¥$^{-1}$ yr^{-1}). In terms of crop yield, CFy of RPDR-Y was the lowest(0.62 kg CO_2 eq kg^{-1} yr^{-1}), while the CFy of RPDR-P was the highest (2.83 kg CO_2 eq kg^{-1} yr^{-1}). Similarly, opposite results were found by different methods in trial 2. The rank of different treatments of CFh, CFy, and CFe were DR-P > DR-Ra > DR > DR-C, DR-Ra > DR > DR-P > DR-C, and DR-Ra > DR > DR > DR-P, respectively. In trials 3, 4, and 5, using different evaluation metrics resulted in different reduction potentials of CF by NT, MS, and RTS compared to a homologous reference treatment. For example, 22.6% of CFh, 38.0% of CFy, and 54.1% of CFe were reduced by MS, respectively, compared to M in trial 4. Although the CF of five agricultural systems in current study was evaluated, it proved that different evaluation metrics of CF provided different results.

4. Discussion

4.1. Carbon sequestration potential and straw return

Increase of SOC and carbon retained in crop biomass are considered eco-system services that diminish the GHG emissions from agricultural systems. Huang et al. (2013) reported that crop biomass could generate a considerable potential for mitigating GHG emissions in the agricultural system, but they suggested considering the long-lived soil carbon pool to estimate the mitigation potential of agriculture systems because the biomass would degrade rapidly.

In current study, annual SOC increase ranged from 476.8 to 13823.9 kg CO_2 eq ha^{-1} yr^{-1} in the five study region except CTWS in trial 5 (Fig. 1), indicating that the RTS in trial 5 could offset the total agricultural GHG emissions from soil and inputs as well as stored CO_2 from the atmosphere in the form of SOC, while the other systems offset 13.8–66.7% of total emissions, resulting in a CF reduction in different crops system (Table 4). Similar results (offset 2–89.3%) were reported in other studies (Huang et al., 2013; Batalla et al., 2015; Li et al., 2016).

Carbon sequestration capacity is easily affected by human activities. Lu et al. (2009) reported that application of N fertilizer, residue return, and no-tillage can respectively sequester 6.0×10^9, 9.8×10^9, and 0.80×10^9 kg C each year in China's crop land. In these techniques, residue return is the most important one. The mitigation potentials of residue return can approximate 34.4×10^9 kg C if this approach is further popularized. In the current study, it is clear that a higher amount of residue return was accompanied by a higher SOCSR, which is concordant with previous results (Lemke et al., 2010; Li et al., 2011b). Without enough residue return, the soil would be a source of CO_2 via loss of SOC, e.g., the CTWS treatments in trial 5 (Fig. 1). Thus, it is important to return considerable residue to maintain positive SOCSR and reduce the CF in agricultural systems.

4.2. GHGs source in agricultural system

In current study, total emissions of soil CH_4, direct and indirect N_2O contributed 840.3–21480.9 kg CO_2 eq ha^{-1} yr^{-1}, while agricultural inputs generated 2698.7–9781.5 kg CO_2 eq ha^{-1} yr^{-1} (Figs. 2 and 3). In uplands, agricultural input was the dominant sector. It has been reported that agricultural inputs could contribute between 69.7% and 96.5% of total GHG emissions in different agricultural systems (Lehuger et al., 2011; Huang et al., 2013). Similarly, agricultural inputs accounted for 65.4–84.8% of total emissions in the current study. In contrast, soil GHGs contributed 55.5–68.7% to total emissions in the paddy soil, which agrees with results reported by Cao et al. (2014) and Wang et al. (2017), whose results showed that soil CH_4 and N_2O (not including indirect N_2O emission) could account for 74.5–85.1% of total GHGs emission. This demanded a different strategy to reduce CF in different agricultural systems.

In the paddy field (trial 2), 94.7–96.5% of soil GHGs was mainly

produced during the rice growing season. To improve agricultural practice, it is important to reduce soil GHGs, especially CH_4. Adaption of super rice may be an effective way to reduce soil GHGs. Wang et al. (2013) reported that total CH_4 emission from super rice field was 35.22% lower than those from traditional rice field. Higher yield was obtained from super rice, and CH_4 emission per kg grain was 81.38% lower than that of traditional rice.

As Figs. 2 and 3 show, N fertilizer was the largest contributor (37.4%–73.2%) to total GHG emissions in uplands, which agrees with the results of other studies (Hillier et al., 2009; Lehuger et al., 2011). Conversely, soil N_2O emission increased linearly with an increasing amount of N fertilizer (Mosier et al., 2006). This indicated that a reduction of N fertilizer could effectively reduce the total GHG emissions from agrochemical productions and associated soil N_2O emissions in different agricultural systems. Increasing N fertilizer applications has been a major management approach to maintain high crop yields. Overuse of N fertilizer by farmers frequently occurs. For example, in the major Chinese cereal rotation systems, annual application rates of N fertilizer are generally more than 500 kg N ha^{-1}. It has been demonstrated that the application rates of N fertilizer can be significantly reduced without decreasing crop yields by adapting optimal nutrient management strategies (Guo et al., 2010). This indicated that the potential of reduction of N fertilizer does exist, thus the decrease of CF of agricultural systems.

Coefficients adapted for calculating GHG emissions of agricultural inputs was another factor affecting the value of CF. We use a set of coefficients reported in 2009 in China (Table 3). In recently years, coefficients for agricultural inputs have been modified, and they are totally dramatically lower than those used in the current study (Wang et al., 2017). For example, the parameters for N, K_2O, herbicides, and pesticides were only 15.0%, 66.3%, 56.5%, and 92.3% of the current study (Liang, 2009; Wang et al., 2017). If the modified coefficients were used, the CF value of different agricultural systems would be greatly decreased. We insist on using the non-modified coefficients because the five study cases were primarily carried out from 2001 to 2011, and the manufacturing technics of inputs of that time would have been relatively under-developed, resulting in higher GHG emissions in that period. The modified coefficients could be used in future studies.

4.3. The effect of evaluation methodology on CF in agricultural systems

4.3.1. CF value highly dependent on functional unit

As a widely used term to evaluate the impact on global climate change, CF was applied with different functional units in varied systems (Gan et al., 2011b; Yang et al., 2014; Al-Mansour and Jejcic, 2016; Toro-Mujica et al., 2017; Vargas-Ramirez et al., 2017; Rashid et al., 2018). Typically, CF was calculated with a functional unit of per unit product in an agricultural system. However, different evaluation methodologies for CF with different functional units in the same system do exist. However, this would probably greatly affect the CF evaluation results. For instance, Knudsen et al. (2014) determined the CF of a four-year rotation system (spring barley-faba bean-potato-winter wheat) under conventional, no input, and slurry scenarios in Jyndevad in Denmark. The results showed that the rank of CF in terms of land surface and crop production were conventional (2599 kg CO_2 eq ha^{-1} yr^{-1}) > slurry (2038 kg CO_2 eq ha^{-1} yr^{-1}) > no input (1453 kg CO_2 eq ha^{-1} yr^{-1}) and no input (695 g CO_2 eq kg^{-1}) > slurry (540 g CO_2 eq kg^{-1}) > conventional (481 g CO_2 eq kg^{-1}), respectively. Similarly, She et al. (2017) reported the CF per unit of the cultivated area of sugar cane (14080 CO_2 eq ha^{-1}) was greater than that of rice (3263 CO_2 eq ha^{-1}). However, its CF per unit of crop yield was approximately a third of the rice in the South China region. Contradictory results also could be found in trial 1 and trial 2 by using these two functional units. The value of CFh of RPDR-Y indicated that it emitted the most GHGs, followed by RPDR-P, among the four treatments, indicating that it is better not to intercrop annual crops in the RPDR system. However, the value of CFy of RPDR-Y showed that it generated the least GHGs while RPDR-P was the highest, indicating that intercropping with higher yield crop (e.g., yam bean) would generate lower CF than RPSR and RPDR (Table 4). Yang et al. (2014) used multiple evaluation metrics to quantitate which was the optimum cropping system among four crop rotation systems in regard to their CF. Although different functional units were used, the results of the CF orderings were consistent. However, this consistent result did not happen among different systems in trial 1 and trial 2 when using similar functional units. Overall, these cases indicated that the CF highly depended on the evaluation methodology used, and different results would suggest different management decisions for reducing CF.

4.3.2. Proper evaluation methodology for CF in agricultural systems

It is suggested that multiple evaluation metrics should be adopted by assessing the environmental influence on different agricultural systems for the purpose of avoiding the one-sided result from the analysis of a single functional unit (Huang et al., 2011a). Recently, Xu et al. (2018) reported the CFs of 19 investigated food products based on different functional units (weight, energy, protein, carbohydrate, and two comprehensive trophic indices: Nu21 and Nu11). Their results showed that the CF values per functional unit varied depending on the functional unit used. To avoid random results, the comprehensive trophic index based functional unit was chosen to illustrate food CFs. As trials 1 and 2 showed, these results indeed avoided a one-sided conclusion (Table 4). However, this raises the question of which approach should be recommended for evaluating the CF of an agricultural system. According to the results of the five trials in current study, we suggest that the CF calculated in terms of economic output is better than those in terms of the other two functional units because the limitation of CFh and CFy was more obvious than CFe. If we calculated CF in terms of per unit area, lower land use intensity would result in lower agricultural inputs, thus it generally produces lower CF (e.g., RPDR-Y and RPDR-P generated more GHG in terms of per hectare than RPSR and RPDR in trial 1). The results neglected the advantage of land saving and would encourage lower land use intensity. Röösa et al. (2013) also reported a similar result in some cases. If we calculated CF in terms of per unit crop product, simply accumulating yield of different crops is probably not logical in complex agricultural systems. For example, the yield of rubber and yam bean were 1253.5 kg ha^{-1} and 7009.1 kg ha^{-1}, respectively. Yam bean in RPDR-Y resulted in the lowest CFy in the four treatments in trial 1 although it had the highest CFh (Table 4). However, rubber (industrial raw materials) and yam bean (generally, a vegetable) were clearly two different materials. Summing up the yield of these two crops seems argumentative. This would be avoided if only each single crop in a complex agricultural system is estimated, such as rice-rice-winter crop one year rotation, maize-wheat one year rotation, as in trial 5. Yet this brings another problem, that is, how to separate the contribution of SOC of each crop in a complex agricultural system. For example, in trials 3 and 4, the dSOC was assessed from the difference of SOC storage between 2004 and 2009, 2008, and 2011, respectively. Determining SOC change in the short term (each crop growing season) is difficult and meaningless because the dynamic of SOC was slow (Triberti et al., 2008), only to estimate the dSOC by added carbon to the soil. Contrarily, when we calculated CF associated with economic output (monetization of crop yield), the limitations mentioned above in terms of per unit area and per unit crop product could be avoided.

Although we suggest that calculating CF values in terms of economic output is superior to other options, it should be recognized that the total quantity of emission is more important from an environmental perspective. Future studies should attempt to design experiments in which the yield of particular crops can be converted such that one unit indicates an equivalent yield of CF. This will combine both the economic and yield approach, and this method may provide better results when compared to single crop yield calculations.

According to the results assessed by CFe, the crop systems in trials 1 and 3 had a lower impact on the environment in terms of global warming potential than trials 2, 4, and 5 (except RTS). A higher impact of global warming potential was recorded by M in trial 4 and CTWS in trial 5 than in the other treatments. A lower N rate application and crop residue return should be adapted to decrease the GHGs induced and manufactured from N fertilizer as well as to enhance mitigation potential by increasing SOC storage. Although five agricultural systems that are typical in China were included in the current study, the dataset for CF calculation was sourced from only one experimental station per agricultural system. It is better to collect a regional database to calculate the CFs of different agricultural systems to generate a representative conclusion in further study.

5. Conclusions

The CF of five agricultural systems with different crop systems in different regions in China was evaluated in terms of CO_2 eq per hectare, per kilogram crop yield, and per unit economic output. The results revealed that multiple evaluation metrics may provide contrary results and hint at a different way to reduce CF in the same crop system. We suggest that it is better to assess CF associated with economic output than the other functional units. According to the results assessed by suggested evaluation metric, the crop systems in agroforestry system and summer maize-winter wheat one-year rotation tillage system had lower impact on the environment in terms of global warming potential than double rice-winter crops one-year rotation system, maize-soybean intercropping system and continuous maize tillage system (except reduced tillage with residue return).

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